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Research Article

Occupancy modeling confirms management concerns for Connecticut (USA) fisher, highlighting the importance of monitoring furbearer distributions

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Abstract

Many mustelids share a history of population decline in North America following European settlement, followed by a period of restoration and recovery. Relative to their historical distributions, mustelids are now perceived to be common and widespread. However, the local conservation status of many mustelids challenges this perception. Fishers (*Pekania pennanti*) in Connecticut were considered abundant prior to the initiation of an annual trapping season in 2005, but indices of relative abundance have decreased over the last 18 years, particularly in western Connecticut. We collected detection data from camera surveys conducted in winter (December–April) in 5 km^2 units (n = 59 units, 354 cameras) throughout Connecticut between 2018 and 2023 to verify management concerns and contribute to an updated conservation assessment of fisher in the state. We applied an occupancy modeling framework to the detection data, emphasizing regional comparisons in occupancy, detection rate, and model-based density. We detected fishers across Connecticut except in the southwestern part of the state, but the area occupied was small (176 km^2). The model-based density estimate (λ) was much lower in Connecticut compared to an abundant fisher population in southern Vermont, and lower in western Connecticut compared to the east. Region was the best-supported covariate on occupancy in our single-season model, and parameter estimates (ψ, p) from top models were greater in eastern Connecticut compared to the west. Fishers did not show spatial segregation from other carnivores in the study but did exhibit temporal separation in daily activity patterns from other carnivores. Our results support management concerns regarding the conservation status of fisher in Connecticut, particularly in the western part of the state. The relative abundance of fisher appears low based on all metrics, and a short-term closure or reduction in the length of the trapping season in western Connecticut could benefit fisher recovery. Harvest data and other sighting metrics may be insufficient to fully assess the conservation status of mustelids such as fisher. Therefore, we recommend implementing a long-term monitoring program to provide current distribution data for fisher. Future studies should focus on understanding the impacts of anticoagulant rodenticides and canine distemper on the current decline of fisher in Connecticut.

Key words: Activity, cameras, carnivore, distribution, region

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Introduction

Furbearers of the family Mustelidae share a legacy of human persecution in North America and habitat loss that followed European settlement, resulting in population declines and range contractions (Laliberte and Ripple 2004; Litvaitis et al. 2006). Protective legislation that regulated annual harvests, along with improved habitat conditions, allowed mustelids to begin recovery and return to at least a portion of their native range (Buckley et al. 2023). Compared to the past, mustelids are now largely considered by managers to be recovered, common, and widespread in North America (Tapper and Reynolds 1996; Northeast Furbearer Resources Technical Committee 2015; White et al. 2015). The Least Concern status of many North American mustelids on the IUCN Red List would seem to support this viewpoint.

The fisher (*Pekania pennanti*) is a good example of a native furbearer species that was once widely distributed throughout the northern portions of North America prior to European settlement, then experienced range-wide declines followed by a period of recovery (Lewis et al. 2012; Powell 1993). In the northeastern United States, fisher were only found in three isolated refugia in the Adirondacks of New York, the White Mountains of New Hampshire, and the Moosehead Plateau of western Maine by the 1930s (Brander and Books 1973; Hapeman et al. 2011). Over the next 60 years, fisher recovered in the Northeast through closed trapping seasons, changes in land-use practices, natural recolonizations, and a series of reintroductions (Douglas and Strickland 1987; Lewis et al. 2012). The largest reintroduction occurred from Maine into Vermont between 1959 and 1967 (Williams et al. 2000) and eventually resulted in a population of fisher distributed throughout the state that has produced an abundant annual harvest, particularly in the northeastern and southern parts of the state (Vermont Fish and Wildlife 2015).

Further south, fisher returned to eastern Connecticut (CT) through natural recolonization from adjacent parts of Massachusetts (Hapeman et al. 2011) but were slow to return to the western part of the state. Movement of fisher into western CT from adjacent New York was limited because they were absent from southern New York until their reintroduction into the Catskills between 1977 and 1979 (New York State Department of Environmental Conservation 2015). To aid in their recovery, fishers (n = 34) were reintroduced to northwestern Connecticut between 1989 and 1991 (Williams et al. 2000). The effort was considered successful, and a follow-up study of the reintroduced population provided information on the effects of season, sex, and site availability on rest-site selection (Kilpatrick and Rego 1994). By 2005, the fisher was considered abundant enough in Connecticut to initiate a statewide trapping season (Rego 2015). Harvests were initially high but decreased by 92% over the next 12 years, along with a catch per unit effort that declined from four to less than one during the same period (Rego 2019). Concurrently, sighting reports and road kills decreased (Rego 2019; Hawley 2022), raising concerns that fishers were declining in Connecticut, particularly in the western part of the state. Eastern and western Connecticut differ in several ways that could be contributing to the observed declines. For example, western Connecticut is characterized by higher elevations, greater snowfall, larger amounts of coniferous forest, and more land classified as human development (NOAA 2023; Perrachio 2020; Rittenhouse

et al. 2022). The smallest percentage of forested habitat in Connecticut also occurs in the southwestern part of the state (Wharton et al. 2004). Potential explanations for the statewide decline include overharvesting, which remains a concern (Fogarty et al. 2022), and ongoing habitat loss (Bridger et al. 2017). Furbearer management is also facing many new challenges, including climate change (Suffice et al. 2017; Fisher et al. 2022), recent changes in mesocarnivore abundance (Litvaitis 2024; Testerman and Hapeman 2022), anticoagulant rodenticides (Gabriel et al. 2012; Buckley et al. 2023; Silveira et al. 2024), and disease (Prince et al. 2021).

One concern is that, despite their range-wide status, many mustelids (e.g., American marten *Martes americana*, wolverine *Gulo gulo*, and mink *Neogale vison*), including fisher, are listed as Vulnerable, Imperiled, or Critically Imperiled (NatureServe) at local levels in states and provinces. Mustelids may not have fully recovered from past declines or may be showing signs of renewed decline (Mowat et al. 2020; Jachowski et al. 2021; Fogarty et al. 2022; Cheeseman et al. 2024). This raises important questions about whether adequate information exists to assess mustelids and whether they should be monitored more closely at local scales. The conservation status of a species is based largely on measures related to abundance and geographical distribution (US Fish and Wildlife Service 1973; IUCN Standards and Petitions Committee 2024). Managers continue to rely heavily on harvest data to provide information about the relative abundance of mustelids because such data can provide valuable insights into population trajectories when corrected for effort (Wilson and Delahay 2001; DeVink et al. 2011). However, managers may only have a coarse understanding of mustelid distributions because of their reliance on data derived from harvested animals, public sightings, and road-kill counts that are not collected using an underlying sampling design. This may explain why locality data from harvested animals appear insufficient to characterize the fine-scale extent of species distributions (Gilbert et al. 2021).

Distribution data are critical for assessing conservation status because estimates such as extent of occurrence and area of occupancy (IUCN 2001) are positively related to population size and are useful when information on population size is unavailable (Keith 1998). Outside of citizen science efforts, distribution data from standardized surveys are rarely collected by managers (White et al. 2015). When such data are available, they may not have been collected frequently enough to accurately reflect current species status, thereby limiting their usefulness for assessing recent conservation trends. Ideally, estimates of mustelid geographical distributions should be based on large-scale monitoring efforts using standardized surveys (e.g., cameras, winter tracks) that produce quantifiable estimates (e.g., extent of occurrence) and are conducted frequently enough to reflect rapid environmental change (Gese et al. 2024). In the absence of adequate distribution data to inform conservation assessments, it remains unclear whether the status of mustelids is sufficiently understood and whether they should continue to be perceived as common and widespread.

A study providing baseline distributional data on fisher in Connecticut and their habitat use from standardized surveys would offer valuable information to verify current management concerns. Combined with regional comparisons of occupancy, detection rate, and model-based density (Royle and Nichols 2003; MacKenzie et al. 2005, 2006; MacKenzie and Royle 2005), such results would

provide the data necessary for a more robust assessment of the current conservation status of fisher in Connecticut. The specific objectives of this study were to (1) provide an estimate of the recent distribution of fisher on public and private lands in Connecticut, (2) compare model-based estimates of density between western and eastern regions of Connecticut, as well as between Connecticut and a southern Vermont population of fisher considered abundant, (3) identify relationships among measured covariates affecting occupancy and detection rate, and (4) examine interactions in activity patterns and co-occurrence between fisher and other mesocarnivores. We predicted that detection rates and occupancy would be higher in eastern Connecticut compared to the west. We also predicted that model-based density estimates would reveal regional differences between Connecticut and Vermont and between eastern and western regions within Connecticut.

Materials and methods

Study area

Camera surveys were carried out throughout the state of Connecticut (USA). The state is characterized by elevations ranging from sea level to the tallest mountain in the western part of the state at 748 m. Winters are cold, with warm, humid summers. The average annual temperature is 10 °C, with 119.94 cm of precipitation as rain and snow. Connecticut is bisected into eastern and western regions by the developed central part of the state, the Connecticut River, and Interstate 91. The landscape is comprised largely of forest, with northern hardwood forest in the northwestern region and the remaining portions of the state occupying part of the Eastern Broadleaf Province (Keys et al. 1995), consisting of oaks (*Quercus* spp.), maples (*Acer* spp.), American beech (*Fagus grandifolia*), and hickory (*Carya* spp.).

Data collection

We collected detection data on fisher from camera surveys in 59, 5 km^2 hexagonal sample units (n = 354 cameras, Fig. 1) over five winter field seasons (December–April) in Connecticut between 2018 and 2023 (Fig. 1). We surveyed each unit only once during the study (units sampled yearly n = 2, 19, 20, 13, 5). We selected a random set of units dominated by forested habitat with at least 70% canopy cover (Allen 1983). Unit size was scaled to 5 km^2, equivalent to the unit size used during a concurrent study of fisher and marten in southern Vermont (Ashbrook and Hapeman 2024; O'Brien et al. 2018). The Vermont study used the same methods described here, which allowed direct comparison of model results between Connecticut and Vermont. The dataset from southern Vermont consisted of 40, 5 km^2 units and was collected during winter months between 2019 and 2022. The population in southern Vermont is considered abundant based on total annual harvest (Vermont Fish and Wildlife 2015) and contrasts with the suspected low-abundance fisher population in Connecticut.

Within each sample unit, we placed an array of six motion-triggered cameras (Bushnell Core DS-4K No Glo cameras; Bushnell Outdoor Products,

Overland, KS) at sites to increase the likelihood of detecting fisher during surveys (O'Connor et al. 2017). Camera locations within units were selected using random sampling, with a minimum distance of 100 m between cameras. At each site, cameras were mounted on trees 1.5–2 m above ground level and aimed toward the base of a bait tree approximately 3–5 m away. On the bait tree, we placed a single can of sardines in water wired approximately 1 m above the ground and emptied another can at the base of the tree. We applied a mixture of skunk essence (Wildlife Control Services, Granby, Connecticut) and petroleum jelly (< 15 ml) to each bait tree and, separately, a curiosity scent ("Marten Magic"; Minnesota Trapping Supplies, Inc., Pennock, Minnesota) containing peppermint–anise oil extracts (< 1.5 ml; O'Brien et al. 2018).

Camera surveys in each unit consisted of 15 days divided into three consecutive 5-day survey periods. Following each 5-day period, we re-scented, re-baited, and exchanged memory cards. We avoided conducting surveys in adjacent units simultaneously to maintain independence. Cameras were active 24 hours per day and were set to no delay with a three-shot photo burst. Images were reviewed independently by a minimum of two people and stored in the Colorado Parks and Wildlife (CPW) Photo Warehouse 4.3 for further analysis. Images classified as unknown species in the CPW database were excluded from analysis if species identification was not possible. Species-specific detection histories recorded during the three survey periods were coded as 0 or 1 (e.g., 010) for each survey within each unit, with a value of 1 indicating detection.

Mapping and distribution

We used detection data from camera surveys to estimate the distribution of fisher in Connecticut by calculating extent of occurrence and area of occupancy following International Union for the Conservation of Nature Red List guidelines (IUCN Standards and Petitions Committee 2024; Kass et al. 2021). The extent of occurrence was estimated by generating a convex hull around peripheral cameras with detections using the Minimum Bounding Geometry tool in ArcPro 3.4.3 (ESRI). This metric provides an estimate of spatial spread, reflecting overall risk from events such as wildfires or disease (IUCN Standards and Petitions Committee 2024). Area of occupancy represents the area actually occupied by the species and was estimated by placing a 2 × 2 km^2 grid over the study area and summing the area of grid cells occupied by fisher. Because we did not sample the entire study area using standard 2 × 2 km^2 grid cells, we also report the proportion of sampled grid cells that were occupied. We did not scale area of occupancy to the same 5 km^2 size used in the occupancy analysis because grid cell size is standardized to 2 × 2 km^2 under Red List guidelines (IUCN Standards and Petitions Committee 2024). Detection rates can be used to index relative abundance given the positive relationship between detection frequency and abundance (Bengsen et al. 2011; O'Brien 2011). Therefore, we mapped detection intensity, defined as the number of cameras within a unit where fisher were detected during the three survey periods (range 0–6), and compared mean values among four defined regions: EastCT (n = 27 units), WestCT (n = 32), AllCT (n = 59), and southern Vermont (n = 40).

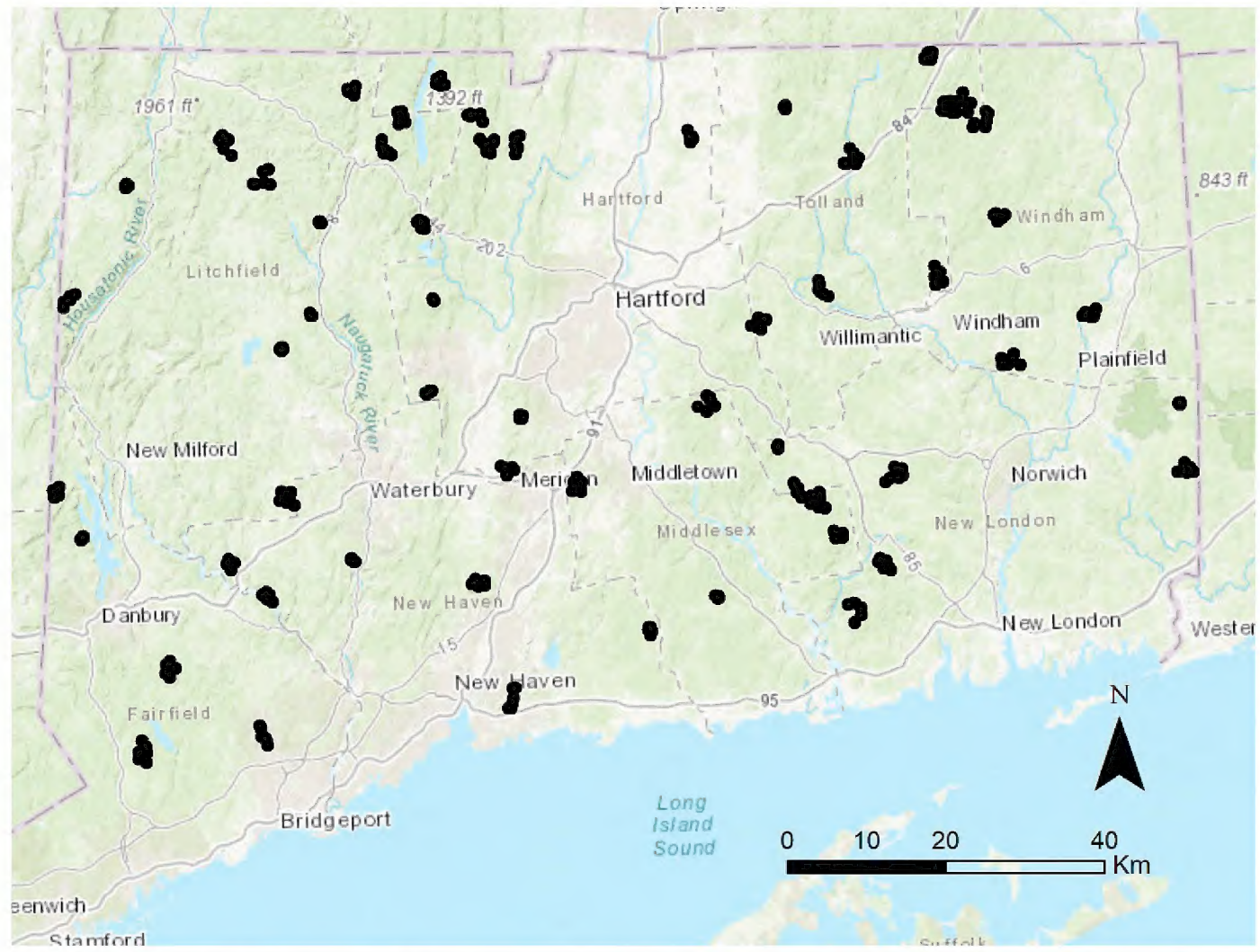

Figure 1. Map showing distribution of camera locations (black dots, n = 354) for fisher (*Pekania pennanti*) during camera surveys conducted between 2018 and 2023 in 5 km^2 units throughout Connecticut.

Covariate data

We identified nine covariates potentially associated with fisher occupancy and detection rate based on previously published studies (Gompper et al. 2016; Manlick et al. 2017; Table 1). Covariate data were collected within sampled units from available GIS layers by extracting values from polygons in ArcPro (ESRI; Table 1). Survey year was included as a covariate on detection rate because data collection spanned multiple years. Collecting detection data across several years could violate the closure assumption of occupancy models, and inclusion of survey year helps account for variation associated with this sampling design. Units were classified as either protected or private land based on the majority land-cover classification within each unit.

Occupancy analyses

We did not assume that detections at cameras within units were independent; therefore, we performed single-season occupancy analyses with covariates at the unit level by pooling detection data from the six cameras within each unit to produce a single detection survey history at the 5 km^2 unit level. For regional comparisons, we produced separate estimates of occupancy and detection rate from base models ($\psi.,p.$) without covariates for datasets containing samples from all of Connecticut, eastern and western Connecticut separately, and Vermont. Regional datasets in Connecticut were created by dividing the state in half, which roughly corresponds to Interstate 91 and the developed central part of Connecticut (Fig. 2). At the time fishers recolonized Connecticut, managers

Table 1. Covariates used in the single-species occupancy analysis of fisher (*Pekania pennanti*) based on camera detection data collected between 2018 and 2023 throughout Connecticut. Parameters refer to occupancy (ψ) and detectability (*p*).

Covariate	Parameter	Prediction	Description
1) Landcover Type e.g., PerDec, PerMix	ψ	Positive for forest types	Percent (Per) of 30 m raster cells within units classified as a specific landcover type[a], including Deciduous (Dec), Mixed, Conifer, Human Disturbance, Wet Woody, and Water.
2) Road_Leng	ψ	Positive	Total linear length of road systems within the unit, road layer from 2024 Tiger files
3) Surv_Yr	ψ, *p*	Positive	Year that surveys were conducted
4) Region	ψ, *p*	Negative	East or west of the center of the state. The center is defined by Interstate 91 that approximately bisects the center of the state.
5) Pub_Priv	ψ	Positive	Whether surveys took place on public or private land
6) Snow_Sur	*p*	Positive	Whether snow was on the ground during the majority of the survey days in a unit
7) Mesocarnivores	ψ, *p*	Negative	Coyote (Coy), fox (Fox), bobcat (Bob), and black bear (Bear)
e.g., FoxOcc, CoyOcc			Site occupancy estimates from base model ψ (.), *p* (.) for mesocarnivores in units[b]
8) Fragmentation e.g., PerPatch, PerCoreLg	ψ, *p*	Negative for more non–core forest	Percentages of units classified as Patch, Edge, and Forest Perforated, or whether a unit has a Core Forest greater than 500 acres derived from the CT Forest Fragmentation Raster layer (https://maps.cteco.uconn.edu/projects/landcover/info/#collapsepanel–w0fleis4e9thl–0–0–03)
9) Total Core Forest	*ψ*	Positive	Total proportion of pixels in unit classified as Core Forest from the Version 2.3 Land Cover layer.

Data sources: **a** – 2021 National Land Cover Database (30 m resolution); **b** – camera surveys.

believed they may have had difficulty moving east to west across the developed central part of the state, prompting the need for reintroduction into northwestern Connecticut.

Next, we used the Royle-Nichols model (Royle and Nichols 2003), which accounts for abundance-induced heterogeneity in detection, to compare mean density (λ) estimates in 5 km^2 units between detection data separated into eastern (n = 27 units) and western Connecticut (n = 32) and between fisher in Connecticut (n = 59) and data collected from the study of fisher in Vermont (n = 40). The Royle-Nichols model assumes that the underlying distribution of abundance can be modeled using a Poisson or negative binomial distribution, closure during surveys, independence of detections among animals, and a constant individual probability of detection at cameras over time. Our focus was to use mean density (λ) estimates as relative density comparisons among data groupings rather than as absolute density estimates.

Many studies have identified interactions between fishers and other carnivores that influence occupancy across their range (Parson et al. 2019; Twining et al. 2024). Therefore, we assessed species interactions among mesocarnivores by first including site occupancy estimates for each mesocarnivore as covariates in single-season, single-species models (Gompper et al. 2016). We produced site occupancy estimates using the program PRESENCE 2.1.3.47 (Hines 2006) for black bear (*Ursus americanus*) and several mesocarnivores (coyote *Canis latrans*, fox *Vulpes vulpes*, *Urocyon cinereoargenteus*, and bobcat *Lynx rufus*) and used them as covariates in models to determine their effects on fisher occupancy and detection rate (Gompper et al. 2016). We then used the two-species conditional occupancy model of Richmond et al. (2010), without covariates, to produce estimates of Δ or φ (Species Interaction Factor, SIF), where values of one indicate that species are detected (Delta Interaction Factor Δ; McHugh et al. 2022) or occupy sites (φ) independently of other species in pairwise comparisons. In each pairwise comparison, fishers were treated as the subordinate predator.

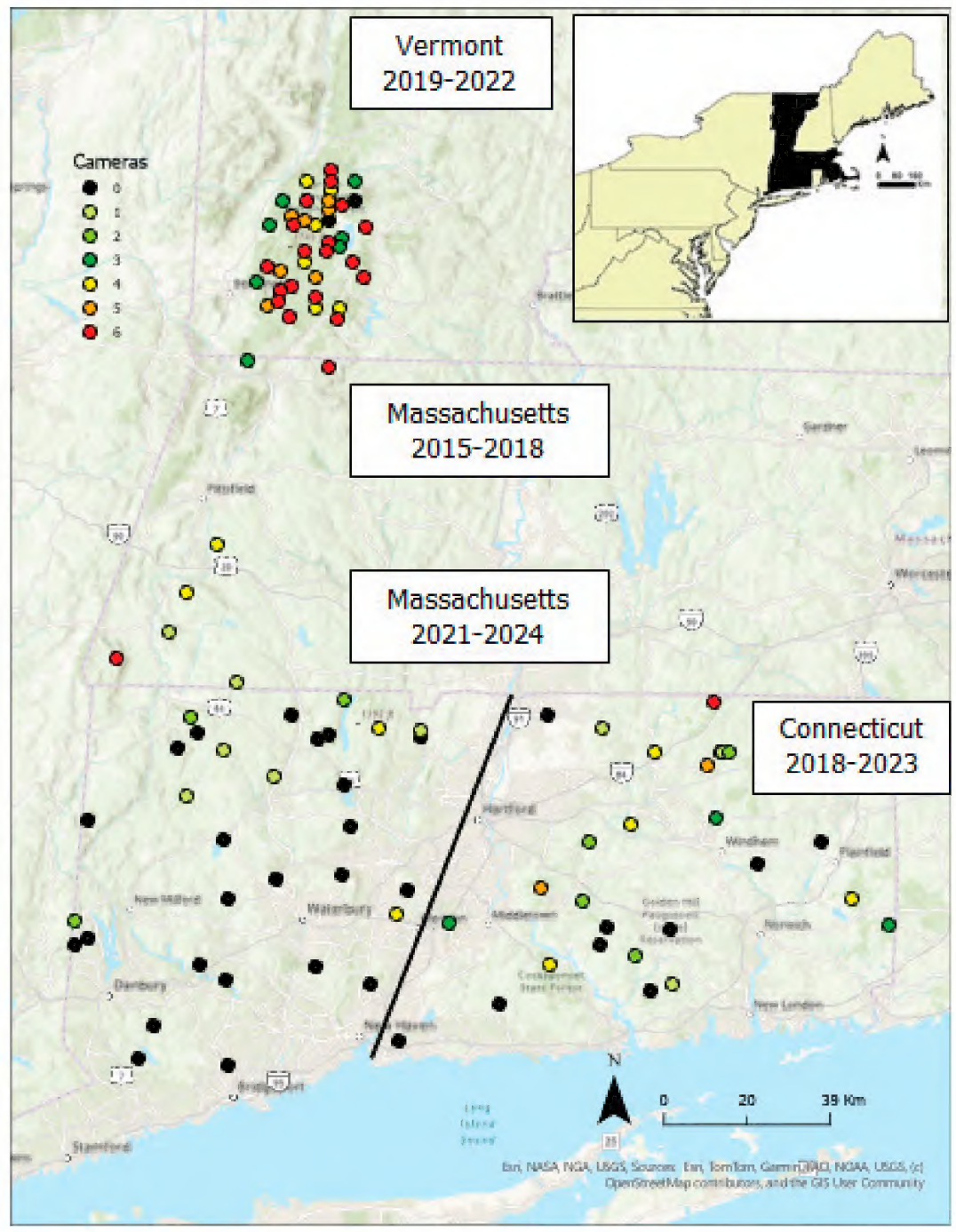

Figure 2. Map showing the number of cameras where fishers (*Pekania pennanti*) were detected during three-week camera surveys in 5 km^2 sampled units from four studies (dates shown on map) spanning Vermont (n = 40), Massachusetts (n = 7), and Connecticut (n = 59). Six cameras were deployed in each unit, and the number of cameras in units with detections (range 0–6) of fisher are distinguished by color. Only samples from Vermont and Connecticut were used in the occupancy-related analyses. The black angled line depicts the I-91 corridor selected as a natural divide between eastern and western Connecticut.

Modeling approach

We used a two-step modeling approach for the single-season analysis, with covariates standardized within MARK (White and Burnham 1999). First, we modeled each focal parameter of interest (i.e., ψ or *p*) with covariates separately while holding the other parameter constant (e.g., ψ (Fox_Occ), *p* (.); Suppl. material 1). We then selected top models in this first stage using Akaike's Information Criterion adjusted for small sample size (AICc; Burnham and Anderson 2002). We applied a relaxed inclusion threshold (ΔAICc < 5) to select top models for each parameter to avoid excluding covariates that may be important when combined with others in more complex models (Morin et al. 2020). Covariates from top models identified in the first stage were then included in a second modeling step. In the second modeling step (Suppl. material 1), we combined covariates in all possible combinations (Doherty et al. 2012) from the top models in the first step to produce an overall set of models that included covariates for combined parameters (e.g., occupancy and detection rate; ψ (Fox_Occ), *p* (Snow_On)). We included no more than four covariates (including

intercepts) to limit overfitting based on sample size (Donovan and Hines 2007) and did not include covariates within the same model when evidence of correlation was present (|r| Pearson > 0.50; Wilson and Schmidt 2015).

We based our interpretations from the single-season analysis only on models in the overall set that were within ΔAICc < 5 of the top model. We used the MacKenzie et al. (2004) goodness-of-fit test in the program PRESENCE (Hines 2006) to examine the fit of each model in the overall set using 1,000 bootstrap replicates. We produced model-averaged estimates for each parameter in MARK using all models in the overall set with ΔAICc < 5 (Buckland et al. 1997). In the overall model set, we established the relative importance of each covariate for each parameter by summing Akaike weights (*w*) across models (ΔAICc < 5) in which each covariate had a significant effect. We considered covariate effects to be significant if the 85% confidence interval (Sutherland et al. 2023) for beta estimates did not include zero.

Activity patterns

We estimated activity pattern overlap among mesocarnivores by collecting detection data at camera sites using 1 hour as the independence interval (Foresman and Pearson 1999). Detection times entered into the CPW database for each species were exported as radians and imported into the "Overlap" package in R version 4.4.1 (Ridout and Linkie 2009) for analysis. We assessed overlap between fisher and black bear, coyote, bobcat, and red fox using the coefficient of overlap (Dhat4 estimator) from Ridout and Linkie (2009). Species with completely overlapping activity patterns have a value of 1, whereas species with no overlap have a value of 0. Differences between activity distributions were considered significant ($P < 0.05$) using the Watson U^2 test conducted in the "CircStats" package (Jammalamadaka and Sengupta 2001) in R version 4.4.1.

Results

Fishers were detected in 27 of 59 sampled units (46%). Nine of 32 unit-level detections (28%) occurred in the western part of the state (9% of total cameras had detections), and 18 of 27 unit-level detections (67%) occurred in the eastern part of the state (33% of total cameras had detections). Notably, fishers were not detected in a large portion of southwestern Connecticut, and detection intensities were greater in the east (mean = 3 cameras per unit where fishers were detected) compared to the west (mean = 2 cameras per unit; Fig. 2). We estimated the extent of occurrence of fishers in Connecticut to be 7,069 km^2, representing 77% of the sampled study area. Area of occupancy was estimated to be 176 km^2 based on the standard 2 × 2 km^2 grid cell size (IUCN Standards and Petitions Committee 2024) and represented 34% of the area occupied by cameras within the study area.

The mean density (λ) estimate of fisher in Connecticut from the abundance-induced heterogeneity model (Royle and Nichols 2003) was 0.72/5 km^2 and was substantially higher in eastern Connecticut compared to western Connecticut (Table 3). The estimate for Connecticut was more than four times lower than the estimate from the sampled fisher population in southern Vermont (3.13/5 km^2).

The single-season analysis of pooled unit-level data yielded four overall models (ΔAICc < 5) with significant covariates on occupancy and detectability (Suppl. material 3). Percent Conifer (sum weight = 0.455) had a positive effect on occupancy, whereas Human Disturbance (sum weight = 0.064) had a negative effect. Fisher had a greater probability of occupancy in eastern Connecticut compared to the west (region; sum weight = 0.481). Fishers were less likely to be detected on private lands (sum weight = 0.735) compared to public lands and were more likely to be detected in eastern Connecticut (region; sum weight = 0.265) compared to the west (Table 2). Models showed some degree of overdispersion, but *p* values were not significant (Table 2). The model-averaged estimate of occupancy for the entire state from the top four single-season models was 0.546 (SE = 0.096, 95% CI = 0.337–0.739), while the mean probability of detecting a fisher during a three-week survey period across the state was 0.557 (SE = 0.067, 95% CI = 0.398–0.706). Occupancy (ψ) of fisher in eastern Connecticut (ψ = 0.694, SE = 0.096, 95% CI = 0.482–0.847) was much higher than in western Connecticut (ψ = 0.329, SE = 0.101, 95% CI = 0.167–0.544). The detection rate (p) of fisher was also substantially higher in the east (p = 0.657, SE = 0.072, 95% CI = 0.505–0.783) compared to the western part of the state (p = 0.475, SE = 0.121, 95% CI = 0.259–0.701). In comparison, the occupancy estimate from the southern Vermont fisher study was 0.951 (SE = 0.035, 95% CI = 0.820–0.988), with an estimated detection rate of 0.909 (SE = 0.028, 95% CI = 0.838–0.951), and fishers were detected on a mean of 5 cameras per unit (Fig. 2).

We collected independent camera detections for the activity pattern analysis for coyote (n = 785), bobcat (n = 60), fisher (n = 184), red fox (n = 243), and black bear (n = 82). Fisher occupancy (φ) and detection (Δ) from two-species co-occurrence models were independent of all other carnivores (Table 4). Values of φ ranged from 0.77 between fisher and bear to 1.02 between fisher and bobcat. Although red fox and bear appeared less likely to co-occur with fisher (φ = 0.76 and 0.77, respectively), and bobcat appeared more likely to co-occur with fisher than expected (φ = 1.02), none of these relationships were statistically supported. Similarly, estimates of Δ ranged from 0.84 between fisher and bear to 1.19

Table 2. The most supported single-species models (ΔAICc < 5) from the program MARK (White and Burnham 1999). *w* = model weight, *k* = number of model parameters. Covariates listed are from the single-species occupancy analysis of fisher (*Pekania pennanti*) based on camera detection data collected between 2018 and 2023 in Connecticut. Region = east or west of Interstate 91; Private = unit classified as public or private land; PerCon and PerHumDist = percentages of land-cover types in each unit derived from the 2021 National Land Cover Database. See Table 1 for detailed descriptions of covariates. c-hat values were derived from 1,000 bootstrap replicates in the program PRESENCE, along with associated *p* values. *Null model included for comparison but not included in model-averaging calculations.

Model	AICc	ΔAICc	w	Likelihood	k	-2 log (L)	c-hat	p Value
ψ (RegEast), *p* (Private)	177.210	0	0.481	1	4	168.469	1.18	0.326
ψ (PerCon), *p* (Private)	178.486	1.275	0.254	0.528	4	169.745	1.22	0.293
ψ (PerCon), *p* (RegEast)	178.957	1.747	0.200	0.417	4	170.216	1.32	0.236
ψ (PerHumDist), *p* (RegEast)	181.262	4.052	0.063	0.131	4	172.522	1.29	0.251
*ψ (.), *p* (.)	188.280	11.069	0.001	0.003	2	184.065	1.52	0.178

Table 3. Model-based estimates of mean density (λ) for fisher (*Pekania pennanti*) derived from the Royle–Nichols (2003) model with abundance-induced heterogeneity implemented in MARK (White and Burnham 1999). Camera detection data used for modeling were collected between 2018 and 2023 throughout Connecticut. *SE* = standard error; *n* = number of 5 km^2 units for each dataset. A total of 95% confidence intervals (LCI, UCI) are included.

Dataset	n	(λ)	SE	LCI	UCI
EastCT	27	1.31	0.40	0.72	2.35
WestCT	32	0.42	0.17	0.19	0.92
AllCT	59	0.72	0.16	0.46	1.12
VT	40	3.13	0.94	1.76	5.57

Table 4. Estimates of φ and Δ from two-species interaction models (Richmond et al. 2010) implemented in MARK (White and Burnham 1999) for fisher (*Pekania pennanti*) and other mesocarnivores in Connecticut. Camera detection data used for modeling were collected between 2018 and 2023 throughout Connecticut. Values of 1 indicate that species are detected (Δ) or occupy sites (φ) independently of other species in pairwise comparisons. *The standard error for φ and the resulting confidence intervals were likely due to coyotes being found in every unit with fisher. SE = standard error and Lower and Upper are the bounds of the 95% confidence interval (UCI, LCI). Bobcat = *Lynx rufus*; red fox = *Vulpes vulpes*; coyote = *Canis latrans*; and black bear = *Ursus americanus*.

Species	φ	SE	LCI	UCI	Δ	SE	UCI	LCI
Bobcat	1.028	0.122	0.932	1.415	1.194	0.283	0.751	1.899
Red Fox	0.764	0.134	0.543	1.078	1.043	0.059	0.932	1.168
Coyote	0.964	*0.000	0.964	0.964	1.002	0.042	0.926	1.089
Black Bear	0.774	0.316	0.347	1.724	0.844	0.205	0.525	1.358

between fisher and bobcat and were not significant (Table 4). We were unable to produce standard errors for φ for fisher and coyote in PRESENCE, likely because coyotes were detected in every surveyed unit.

Daily activity analyses showed that fisher, fox, and coyote were strongly nocturnal, whereas bobcat and bear exhibited a greater proportion of activity during daylight hours (Fig. 3). Overall activity overlap among carnivores was generally low (Dhat4 = 0.344–0.629; Fig. 3), but higher overlap was observed between fisher and fox, fisher and coyote, and fox and coyote (Dhat4 = 0.774–0.787; Suppl. material 2).

Discussion

The results from this study support management agency concerns regarding the conservation status of fisher in Connecticut, particularly in the western part of the state. Our estimate of extent of occurrence indicates that fishers are widespread throughout the state except in southwestern Connecticut, but the area of occupancy estimate indicates that they are using only a small portion of that area. All parameter estimates (e.g., ψ, *p*, and λ) were greater in the eastern part of the state compared to the west (Fig. 2). Fishers were 5.75 times more likely to occupy eastern Connecticut compared to the west and 4.76 times more likely to be detected there.

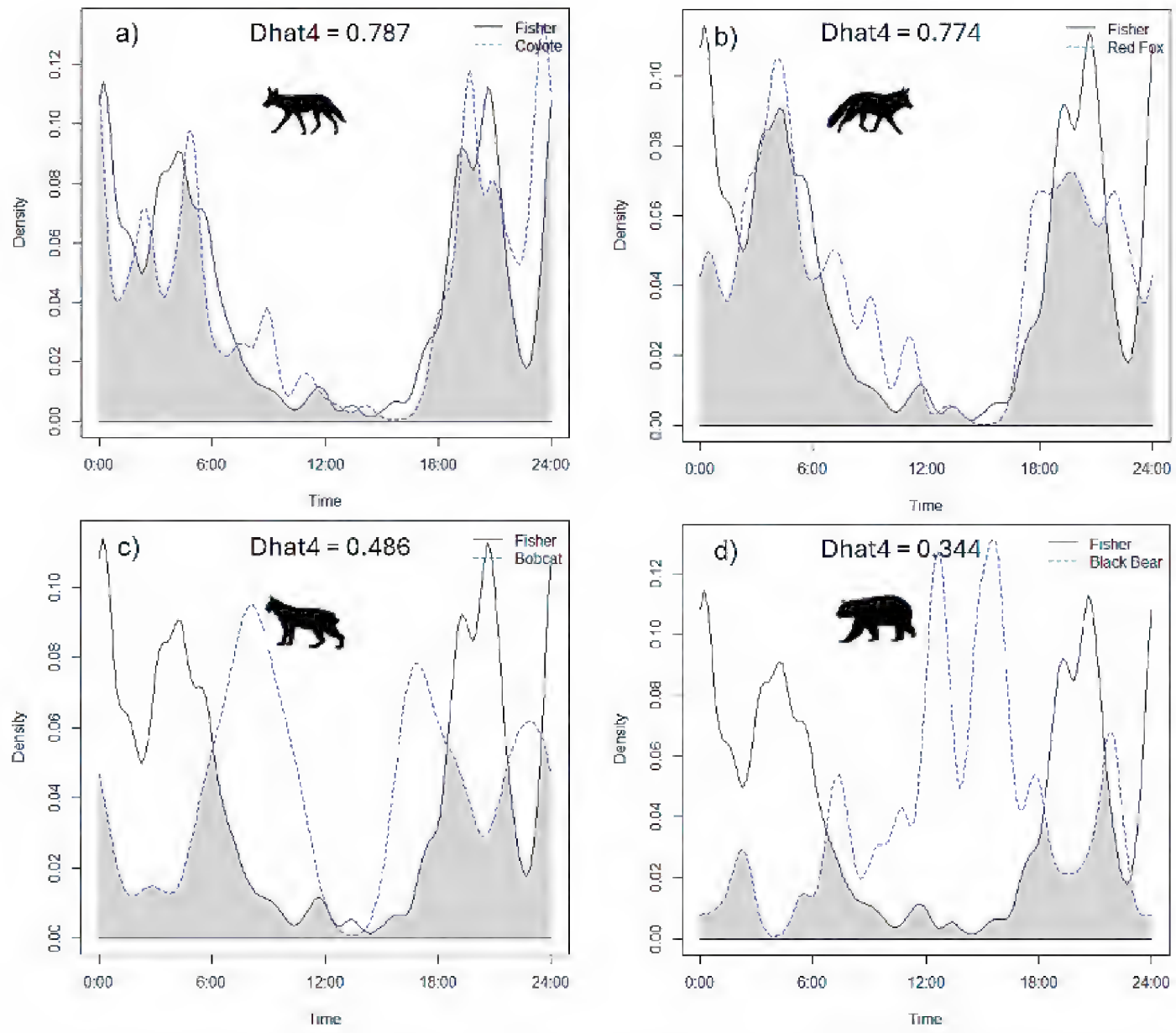

Figure 3. Measures of daily activity overlap (Dhat4; Ridout and Linkie 2009) from the "Overlap" package (Meredith and Ridout 2024) in R version 4.4.1. Values of Dhat4 range from 0 to 1, with higher values indicating greater overlap in activity. Comparisons include **a.** Fisher and coyote; **b.** Fisher and red fox; **c.** Fisher and bobcat, and **d.** Fisher and black bear.

Differences in the distribution of fishers between eastern and western Connecticut could be associated with the degree of isolation from other fisher populations outside the state. Fishers returned to eastern Connecticut around the late 20th century through natural recolonization from Massachusetts and possibly Rhode Island (Hapeman et al. 2011), and presumably that connectivity remains today. Population expansion from eastern to western Connecticut may have been impeded by natural barriers, including the Connecticut River, and an anthropogenic barrier, the I-91 corridor, which is comprised largely of human development. This was likely a contributing factor for the reintroduction of fishers to the northwestern part of the state between 1989 and 1991 (Kilpatrick and Rego 1994). Movement of fishers into western Connecticut from adjacent New York was limited because they were absent from southern New York until their reintroduction into the Catskills between 1977 and 1979 (New York State Department of Environmental Conservation 2015). Fisher harvest in New York remains relatively low in southern New York compared to the northern portion of the state (New York State Department of Environmental Conservation 2021, 2024), and few fishers have been harvested in the Southern Taconic Highland management unit immediately adjacent to western Connecticut (New York State Department of Environmental Conservation 2024). Genetic data suggest the possibility that fishers moved south into western Massachusetts and Connecticut following their reintroduction to Vermont between 1959 and 1967 (Hapeman et al. 2011). However, preliminary data from our ongoing cam-

era surveys in western Massachusetts (2021–2024) have revealed a potential latitudinal gradient of decreasing detection intensities of fisher from southern Vermont through western Massachusetts and into northwestern Connecticut (Fig. 2). If this pattern is supported with additional data, it could indicate limited movement of fishers into western Connecticut from western Massachusetts.

Several environmental and topographic features (e.g., elevation, snow levels) differ between eastern and western Connecticut. Of the variables we measured, we identified significant associations between occupancy and the amount of conifer habitat and human disturbance. Fishers were more likely to occupy units with greater amounts of coniferous habitat, and related studies have highlighted the importance of conifer habitats for fisher in the eastern United States (Arthur 1989; New York State Department of Environmental Conservation 2015; Fuller et al. 2016). Fishers are considered specialists for resting habitat but more generalist when moving and foraging (Olson et al. 2024). When conifer habitats are available, fishers may preferentially use them over hardwood forests for resting and maternal denning opportunities (Powell 1994; Powell et al. 1997). Coniferous habitat did not explain differences in distribution between eastern and western Connecticut because fisher had lower occupancy in the west, where there is a greater amount of conifer habitat. Fishers were less likely to occupy units with higher levels of human development, which is consistent with the lower occupancy observed in western Connecticut, particularly in the southwest, where development is more extensive. Fisher occupancy is negatively associated with road density (New York State Department of Environmental Conservation 2015; Ellington et al. 2017), but fishers can inhabit developed areas by altering activity and behavioral patterns in response to human disturbance (Ganoe et al. 2024b, LaPoint et al. 2013). Fishers in developed areas may require core areas with dense canopy cover as refugia to limit exposure to human presence (Kordosky et al. 2021) and sufficient habitat, even if suboptimal, to connect existing habitat patches (LaPoint et al. 2013). Higher detection rates on public compared to private land did not explain differences in fisher distribution between eastern and western Connecticut and were contrary to our expectations. Many private properties surveyed had limited public access and showed fewer signs of human disturbance compared to the public properties sampled. Detection rates on public lands may have been greater because these lands tended to consist of larger, contiguous habitat blocks that could support larger numbers of individuals.

By 2005, managers considered the fisher population in Connecticut abundant enough to initiate an annual harvest; yet 18 years later, the annual harvest has decreased by more than 95% (Hawley 2022). All available metrics indexing relative abundance, including harvest levels, sightings, and roadkill counts, suggest that fishers have declined in Connecticut. Density estimates from this study were much lower, particularly in western Connecticut, compared to the fisher population in southern Vermont that is considered abundant based on long-term harvest data (Table 3). We recognize that our density estimate represents a single snapshot in time and that additional estimates from the past or future are needed to provide context and draw more robust conclusions regarding population trends in Connecticut. We believe that assumptions of the Royle-Nichols model were reasonably met, and although camera surveys within units satisfied the closure assumption based on the short 15-day survey win-

dow, we acknowledge that fishers may have moved in and out of units across the multiple years of sampling. In light of this, we interpreted density estimates as relative comparisons rather than absolute values, as recommended by Gilbert et al. (2021). Overall, available evidence indicates that the fisher population has declined in Connecticut, consistent with other recently reported declines across their range, including Rhode Island (Ganoe et al. 2024a), northern Wisconsin (Kirby et al. 2018), New Hampshire, and Minnesota (Litvaitis 2024).

The cumulative impacts of multiple stressors can ultimately lead to population decline (Tyack et al. 2022). One primary stressor for fisher in Connecticut is overharvest following initiation of the 2005 trapping season. Harvest numbers and catch per unit effort began to decline immediately after the onset of the annual harvest, and this trend has continued (Hawley 2022). Sighting and vehicle-kill reports, which increased between 2000 and 2005, declined by 89% in the years following initiation of the annual trapping season (Rego 2019). The initial bag limit of two in 2005 was increased to four in 2010, reduced back to two in 2019 (Rego 2019), and finally lowered to one in 2022 (Hawley 2022). The combined declines in harvest success, fisher sightings, and roadkill reports suggest that trapping seasons with bag limits greater than two were not sustainable. Fishers are susceptible to overharvest (Proulx 2000), which was a principal factor contributing to their historical decline across North America following European settlement (Lewis and Zielinski 1996; Powell 1979, 1993; Williams et al. 2007; Buskirk et al. 2012). A recent example from British Columbia confirms that overharvest remains a conservation concern for fisher management (Fogarty et al. 2022). Although the impact of annual harvest in Connecticut has likely diminished in recent years due to reduced trapper participation and lower bag limits, relative fisher density remains low. Therefore, additional stressors may be contributing to the continued low abundance of fisher in Connecticut and warrant further investigation.

Multiple anticoagulant rodenticides that can cause direct and indirect mortality in animal populations (Gabriel et al. 2012) have recently been detected in harvested fishers from Vermont, New Hampshire, and Maine (Needle et al. 2019; Buckley et al. 2023; Silveira et al. 2024). At this time, there is insufficient information to assess their impact on fisher in Connecticut. Mustelids, like other carnivores, are also susceptible to canine distemper (Terio and Craft 2013; Sweitzer et al. 2015). Canine distemper has been reported in Connecticut in recent years in raccoons (*Procyon lotor*) and skunks (*Mephitis mephitis*), raising concerns for fisher that can interact with both species (J. Hawley, Department of Energy and Environmental Protection, personal communication). Fluctuations in fisher populations have been associated with mast cycles of hardwood species such as oak (*Quercus* spp.), American beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), and mountain ash (*Sorbus americana*; Jensen et al. 2012). Mast production can influence fisher productivity directly because it is a regular component of their diet. Indirectly, mast production influences the abundance of small mammals that fishers feed on and can improve juvenile survival during years when small mammal populations increase (Jensen et al. 2012). Abundant mast years have been shown to reduce trap vulnerability of fishers in the Adirondacks of New York, which in turn reduces harvest rates (Jensen et al. 2012). Acorn survey results in Connecticut during the period of our camera surveys showed a cyclical trend (Barsky 2023), with low production for red oaks in 2018 and white oaks in 2019, but a bumper crop for red oaks in 2019 (Connecticut Agricultural Experi-

ment Station 2024). Several years of spongy moth (*Lymantria dispar*) outbreaks have likely affected acorn production and mortality during the study period (Connecticut Agricultural Experiment Station 2022), but relationships among mast production, spongy moths, and fishers in Connecticut require further study.

Mesocarnivores such as bobcats and coyotes have shown recent increases in Connecticut and could be negatively affecting fisher. Bobcats and coyotes are known to compete with and prey on fisher in other parts of their range (Gabriel et al. 2015; Sweitzer et al. 2015; Wengert et al. 2013). Fishers in this study did not spatially segregate from other carnivores examined. A similar finding was reported for fisher and bobcat in Wisconsin in the presence of several types of competitive interactions (Gilbert 2000). Fishers did exhibit temporal separation in daily activity patterns from other carnivores, which could reduce competition or predation risk. Alternatively, the low level of daily activity overlap between fisher and bobcat may simply reflect the greater amount of diurnal activity by bobcats compared to fishers (Ganoe et al. 2024b; Hubbard et al. 2022). The overall impact of mesocarnivores on fisher may not have been detected using an occupancy-based approach. Lonsinger (2022) concluded that violating the closure assumption can limit detection of avoidance or aggregation in co-occurrence models. It is possible that no patterns were detected because closure assumptions were violated during surveys conducted over five winter seasons. However, the short survey duration of 15 days and the relatively large sample units (5 km^2) compared to fisher movement rates are not consistent with survey designs that fail to meet closure assumptions (Lonsinger 2022). We also used more than one method to assess co-occurrence patterns, and neither produced significant results. It is possible that coyotes and bobcats are influencing vital rates of fisher (e.g., dispersal, survivorship) that were not measured in this study, and this warrants further investigation. A similar conclusion was reported for American marten (*Martes americana*) in the Green Mountains of southern Vermont (Ashbrook and Hapeman 2024).

Several important considerations relate to our study design and inferences. We collected data over several years to complete the dataset, which could have violated the closure assumption if individuals moved in or out of survey areas. However, surveys within individual units were short enough to meet closure assumptions of occupancy models, and we accounted for interannual variation in detection rates to address multi-year data collection. It remains possible that closure assumptions were violated; therefore, we interpret occupancy estimates as probabilities of site use and apply inferences from parameter estimates to the full sampling period rather than a single year. For similar reasons, density estimates from the Royle-Nichols model were interpreted as relative measures rather than absolute values, as recommended by Gilbert et al. (2021). Surveys were conducted during winter in areas predominantly classified as forested, with high levels of leaf-on canopy cover (Allen 1983), and used a 5 km^2 sampling unit. Detection rates may differ if surveys were conducted during summer, and fishers in Connecticut may occur in smaller forest patches or other land-cover types not sampled. Additional year-round surveys at finer spatial scales could provide a more complete estimate of the fisher distribution. Finally, we were unable to include all covariates that may explain differences between fisher in eastern and western Connecticut, such as availability of den trees, relative abundance and diversity of food resources, and patch size and configuration in fragmented landscapes.

Conclusion

The conservation status of a species is based on criteria used to evaluate information about population size, population change, and geographical distribution. Prior to this study, no standardized surveys for fisher had been conducted in Connecticut to assess their statewide distribution. The results of this study are broadly applicable to other harvested mustelids, such as American marten (*Martes americana*) and mink (*Neogale vison*), and highlight the need for closer monitoring of these species through regular collection of distribution data using standardized methods. Harvest data, along with other sighting metrics, may be insufficient to assess conservation status. Long-term monitoring of fisher distribution in Connecticut will help establish baselines, identify changes over time, inform management agencies, and provide a more complete estimate of distribution. Fishers in Connecticut are currently listed as Imperiled by NatureServe, and a recent proposal to elevate their status to a Species of Greatest Conservation Need has been included in the state's latest Wildlife Action Plan based on documented declines and low detection across multiple metrics. We acknowledge that the current status of fisher in Connecticut likely reflects multiple stressors, including past over-harvest. Fishers in western Connecticut are not abundant, and additional trapping restrictions in this region (e.g., closing the trapping season or reducing season length) could aid recovery. Fishers have demonstrated the capacity to recover quickly when trapping pressure is reduced. A closed trapping season in New York between 1936 and 1948 allowed fisher populations to recover, and harvest resumed by 1949 (Powell 1993). We support further data collection on the presence of anticoagulant rodenticides and canine distemper in fisher populations in Connecticut, given recent work highlighting their prevalence in fishers in the Northeast (Silveira et al. 2024) and recent detection of canine distemper in other mesocarnivores. Finally, we recommend collecting additional data (e.g., camera-based and genetic data) to better evaluate connectivity between fisher populations in western Connecticut and those in adjacent areas of New York and Massachusetts.

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Additional information

Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

Camera surveys were conducted under a state permit (AU21p) issued by the Connecticut Department of Energy and Environmental Protection.

Use of AI

No use of AI was reported.

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Author contributions

PH conceived the ideas and designed the methodology; KG and PH collected the data; PH and KG analyzed the data; and PH led manuscript writing. All authors contributed to manuscript review and approved the final version for publication.

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Data availability

All of the data that support the findings of this study are available in the main text or Supplementary Information.

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Supplementary material 1

Stepwise modeling approach for single-season occupancy models of fisher (*Pekania pennanti*) with covariates

Authors: Paul Hapeman, Katerina Gillis

Data type: pdf

Explanation note: The dataset consisted of 354 cameras in 59, 5 km2 units collected during camera surveys between 2018 and 2023 throughout Connecticut.

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Link: https://doi.org/10.3897/natureconservation.61.168852.suppl1

Supplementary material 2

Measures of daily activity overlap

Authors: Paul Hapeman, Katerina Gillis

Data type: docx

Explanation note: Measures of daily activity overlap (Dhat4; Ridout and Linkie 2009) derived from the "Overlap" package (Meredith and Ridout 2014) in R. Values range from 0 to 1, with higher values indicating greater overlap in activity. Independent detections used for the analysis were coyote (n = 785), bobcat (n = 60), fisher (n = 184), red fox (n = 243), and black bear (n = 82). Camera detection data used for modeling were collected between 2018 and 2023 throughout Connecticut. *Differences between activity distributions were considered significant ($P < 0.05$) using the Watson U^2 test implemented in the "CircStats" package (Jammalamadaka and SenGupta 2001) in R version 4.4.1.

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Link: https://doi.org/10.3897/natureconservation.61.168852.suppl2

Supplementary material 3

Beta estimates of covariates from top four models

Authors: Paul Hapeman, Katerina Gillis

Data type: docx

Explanation note: Beta estimates of covariates from top four models (see Table 1) from the single season occupancy analysis of fisher (*Pekania pennanti*). Camera detection data for the modeling was collected between 2018 and 2023 throughout Connecticut. 85% confidence intervals (LCI, UCI) were included based on the suggestion from Sutherland et al. (2023) and relationships of covariates were significant if confidence interval for beta estimate did not include 0. Relationships of covariates labeled positive (+) or negative (−).

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Link: https://doi.org/10.3897/natureconservation.61.168852.suppl3

Supplementary material 4

Detection information for fisher in Connecticut

Authors: Paul Hapeman, Katerina Gillis

Data type: xlsx

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